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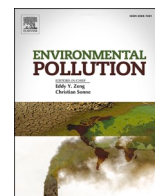
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Changes in air quality in Mexico City, London and Delhi in response to various stages and levels of lockdowns and easing of restrictions during COVID-19 pandemic^{☆,☆☆}

E. Vega^{a,*}, A. Namdeo^b, L. Bramwell^b, Y. Miquelajauregui^c, C.G. Resendiz-Martinez^d, M. Jaimes-Palomera^d, F. Luna-Falfan^d, A. Terrazas-Ahumada^d, K.J. Maji^b, J. Entwistle^b, J. C. Núñez Enríquez^e, J.M. Mejia^e, A. Portas^f, L. Hayes^g, R. McNally^g

^a Centro de Ciencias de la Atmósfera, National Autonomous University of Mexico (UNAM), Mexico

^b Geography and Environmental Sciences Department, Northumbria University, UK

^c Laboratorio Nacional de Ciencias de la Sostenibilidad, Instituto de Ecología, UNAM, Mexico

^d Secretaría de Medio Ambiente, Dirección de Monitoreo de Calidad del Aire, Subdirección de Análisis y Modelación, Mexico

^e Instituto Mexicano del Seguro Social (IMSS), Mexico

^f Mathematics, Physics and Electrical Engineering, Northumbria University, UK

^g Population Health Sciences Institute, Newcastle University, UK

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ABSTRACT

The impacts of COVID-19 lockdown restrictions have provided a valuable global experiment into the extent of improvements in air quality possible with reductions in vehicle movements. Mexico City, London and Delhi all share the problem of air quality failing WHO guideline limits, each with unique situations and influencing factors. We determine, discuss and compare the air quality changes across these cities during the COVID-19, to understand how the findings may support future improvements in their air quality and associated health of citizens. We analysed ground-level PM₁₀, PM_{2.5}, NO₂, O₃ and CO changes in each city for the period 1st January to August 31, 2020 under different phases of lockdown, with respect to daily average concentrations over the same period for 2017 to 2019. We found major reductions in PM₁₀, PM_{2.5}, NO₂ and CO across the three cities for the lockdown phases and increases in O₃ in London and Mexico City but not Delhi. The differences were due to the O₃ production criteria across the cities, for Delhi production depends on the VOC-limited photochemical regime. Levels of reductions were commensurate with the degree of lockdown. In Mexico City, the greatest reduction in measured concentration was in CO in the initial lockdown phase (40%), in London the greatest decrease was for NO₂ in the later part of the lockdown (49%), and in Delhi the greatest decrease was in PM₁₀, and PM_{2.5} in the initial lockdown phase (61% and 50%, respectively). Reduction in pollutant concentrations agreed with reductions in vehicle movements. In the initial lockdown phase vehicle movements reduced by up to 59% in Mexico City and 63% in London. The cities demonstrated a range of air quality changes in their differing geographical areas and land use types. Local meteorology and pollution events, such as forest fires, also impacted the results.

1. Introduction

The World Health Organization (WHO) declared a pandemic after the outbreak of a novel coronavirus 2019 (COVID-19) infectious disease in the People's Republic of China in December 2019 (Huang et al.,

2020). Among the main recommendations by the WHO, it was advised that all countries should be prepared for containment, isolation, and case management, contact tracing and prevention. Consequently, most countries imposed strict measures to minimize the spread of the infection. Countries and cities around the world adopted varying levels of

[☆] This paper has been recommended for acceptance by Da Chen. ^{☆☆} This study has demonstrated the level of reductions in activity needed to achieve air quality targets.

^{*} Corresponding author.

E-mail address: evega@atmosfera.unam.mx (E. Vega).

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lockdowns in the initial stages of COVID-19 spread. Most countries implemented population confinement or 'lockdown', which resulted in a significant reduction in the number of vehicle movements, goods transportation, and industrial processes. This triggered industrial, commercial, and financial slowdowns. It has been suggested that air pollution particles may act as vehicles for viral transmission and may be associated with increased COVID-19 mortality (Gerretsen, 2020; Setti et al., 2020; Yongjian et al., 2020). In contrast, it has been found that a high concentration of PM did not play a key role in COVID-19 spread (Collivignarelli et al., 2021; Belosi et al., 2021).

1.1. City pollutant characterization

Mexico City was once the most polluted mega-city worldwide, but after the implementation of several policies, for more than three decades, there has been a significant achievement in reducing air pollution exposures to the population. Despite air quality improvements, current PM_{2.5} and O₃ levels in Mexico City are frequently above Mexican air quality standards (Sedema, 2020). Coal-burning was historically the dominant contributor to London's air pollution. Environmental regulation, heavy industry moving out of the city, and switching energy sources made huge improvements to London's air quality since the 1950s (Fuller, 2018). In 2001, the use of diesel vehicles was encouraged to reduce greenhouse gases as their CO₂ emissions are lower than petrol cars. Unfortunately, diesel emissions were higher in NO₂ and particulate matter, so now there are policies in place for phasing out diesel vehicles. Vehicle traffic is the major polluter of London's air (Greater London Authority, 2020), and despite multiple clean air zone interventions reducing vehicle use, it still fails to meet international standards for air quality. Thirteen Indian cities are among the top 20 most polluted cities worldwide, with Delhi being the 6th most polluted city (PM_{2.5}: annual average of 143 µg/m³) (World Health Organization, 2018). In winter 2018, the annual average PM_{2.5} concentration, reported by four air quality monitoring stations (Anand Vihar, Punjabi Bagh, RK Puram and Okhla) located across the city, was above 300 µg/m³, which is approximately 5 times higher than the Indian NAAQS of 60 µg/m³, and 12 times higher than the WHO guideline of 25 µg/m³ (Nandi, 2018). The hazardous level of pollution concentration was due to various factors including road dust, vehicle pollution, brick kilns, informal small industries and cold weather (Amann et al., 2017). Delhi's pollution problem is also caused by agriculture when harmful particles are produced by burning crop stubble in neighbouring states.

1.2. Restriction characteristics

Approaches to the implementation and easing of COVID-19 related restrictions have varied across the world and produced varying air quality responses. Air quality reduced as air polluting activities began to return (Baldasano, 2020; Tobías et al., 2020; Rodríguez-Urrego et al., 2020; Xiang et al., 2020; Wyche et al., 2021). In Mexico City, measures to stop the spread of the virus by restriction of citizen mobility began on March 14th, 2020 (Table A1). This included a social distancing campaign and the closure of all educational institutions. On March 24th non-essential activities, schools and universities were closed and gatherings banned. A state of emergency was declared on March 30th, to reduce the load on the medical care assistance. These strict restrictions were eased from May 31, 2020. In London, restrictions came into force on 24th March with the public required to stay at home. From June 1st services were slowly reinstated and some school classes returned. The lockdown in Delhi began on 25th March when all but essential sectors closed. From June 1st activity returned to normal except for some restrictions on movement from 9 p.m. to 5 a.m. Further details of lockdown restrictions and phases are provided in Table S11. The instruction to stay at home could be expected to decrease traffic volume, followed by an increase associated with the easing of the lockdown restrictions. Road traffic emissions have shown to be a major source of air pollution,

particularly in North America and Europe (Comert, 2020; Baldasano, 2020; Kelly, 2011). The reduction of traffic is commonly associated with improved air quality due to the reduction in some pollutants, as well as health improvements where disease, such as asthma, is aggravated by these pollutants (Matz et al., 2019; Nori-Sarma et al., 2021).

Mexico City lies within the Mexico City Metropolitan Area (MCMA), a megalopolis with over 20 million inhabitants and a vehicular fleet of 5.7 million (INEGI, 2020). In Mexico City the percentage reduction in vehicular traffic registered was 59%, 70% and 49%, in the two lockdowns then unlock phases respectively (Semovi, 2020). Reduced activity and emissions from industry and the service sector also reduced air pollution levels. Each of the included cities have distinctly different weather systems impacting their air quality. Meteorology in Mexico City is strongly influenced by its high altitude. It is a dry region with an average temperature of 18 °C and higher temperatures during the warm dry season from April to May. The population is exposed to poorer air quality during the cool dry season (November to February) with higher O₃ and suspended particle concentrations, especially in the mornings (Sedema, 2020). London is the largest city in the UK with a population of approximately 9 million. London's NO₂ emissions are minimised by congestion charges and low and ultra-low emissions zones. London has a temperate maritime climate with warm summers (June to August), cool winters (December to February), an annual average temperature of 11 °C, year-round rain and frequent moderate to strong winds most usually from the west (average wind speed 4.7 m/s) (Weatherspark, 2020). O₃ levels are highest in spring and summer in London when conditions are sunny with low wind speeds (London Air, 2020). London usually has poorer air quality in the winter months when temperature inversions trap pollutants at ground level. Road transport is the primary source of air pollution in London although some domestic solid fuel burning does still add to this in colder months (Jephcote et al., 2021).

Delhi, the capital of India, is the second-largest megacity in the world and the largest urban agglomeration in this country with an estimated population of 19.3 million in 2020 (Census, 2011). The overall population density is 11,297/km². It is located at an elevation of 216 m above the mean sea level (Census, 2011). In March 2018, Delhi had 10.8 million registered vehicles, including 6.96 million motorcycle/scooter and 3.1 million motor-car (private vehicles) (Transport Department Government of NCT of Delhi, 2018). In Delhi, closure of brick kilns and construction sites and significant reduction of traffic movement has improved air quality during lockdown phases, although, the effect of the lockdown was found to be less pronounced on the sources like secondary chloride, power plants, dust-related, hydrocarbon-like organic aerosols (HOA), and biomass burning related emissions (Manchanda et al., 2021). Delhi has a semi-arid climate and is surrounded by the mountain region of the Himalaya to the north, central hot peninsular region to the south, hilly region to the east and, to the west the Great Indian Desert. Delhi experiences four main seasons: winter (December–February), summer (March–May), monsoon (June–August) and post-monsoon (September–November). Temperatures range between 7 ± 3 °C in winter and 45 ± 3 °C in summer (Kumar et al., 2020). Crop stubble burning in nearby states during October has a major detrimental impact on Delhi's air quality (Amann et al., 2017).

In this article, we analysed air quality changes for the period 1st January to August 31, 2020 in comparison with the average daily concentrations over the same months for 2017 to 2019 in Mexico City and London and 2017 to 2018 for Delhi. The three megacities were selected to incorporate varying geopolitical circumstances. First, the three cities are situated on different continents (America, Europe and Asia) and experience different climates. The latter provides insights into potential differences in meteorological and air quality interactions. Second, analyzing air quality across the three cities provide information on how air quality management, partly associated with different socio-economic contexts such as national income (i.e. low middle income in India, high middle income in Mexico, and high income -UK countries) could affect air quality. We examined and compared factors impacting air quality in

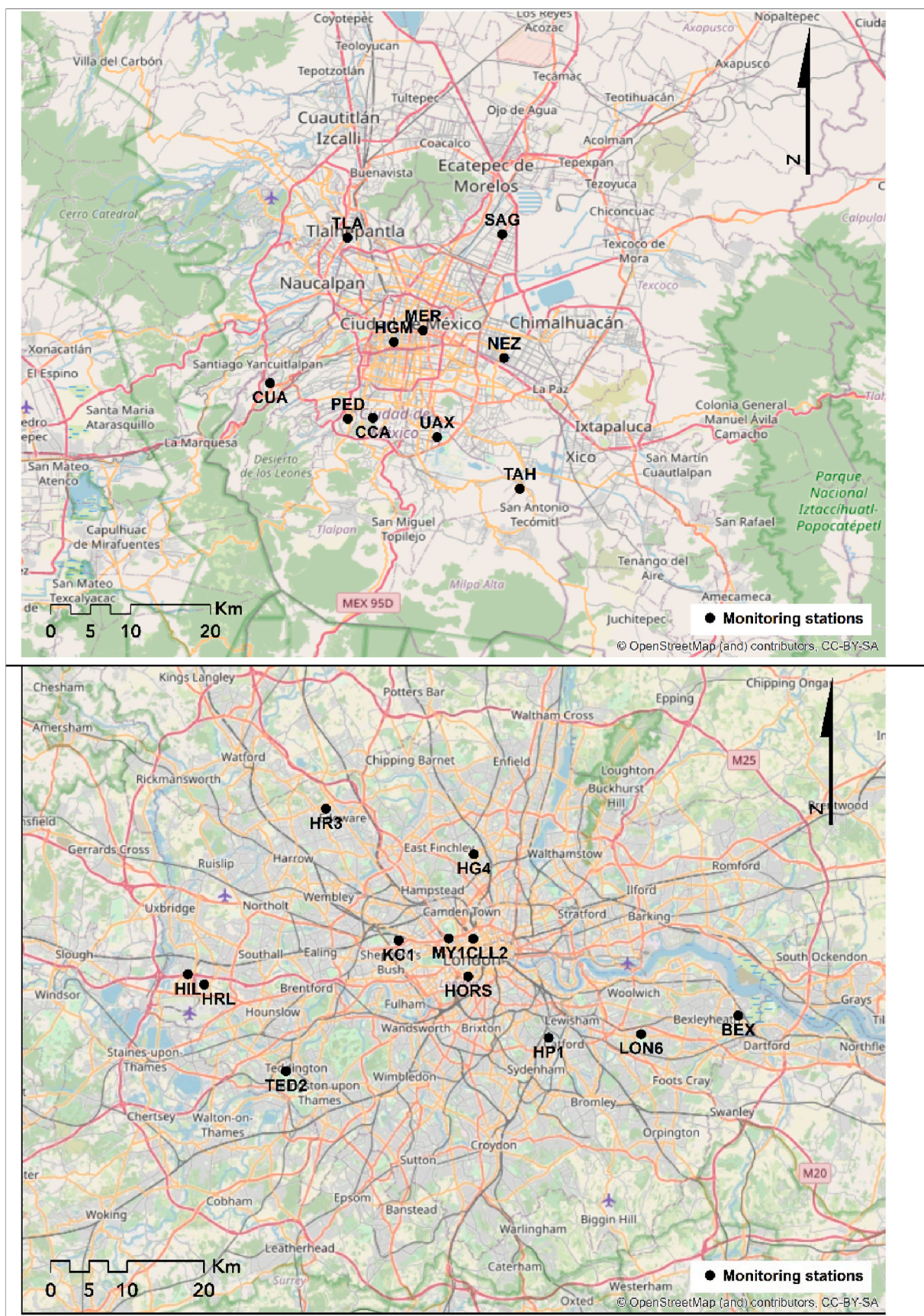


Fig. 1. Maps showing locations of air quality monitoring stations in Mexico City, London, and Delhi.

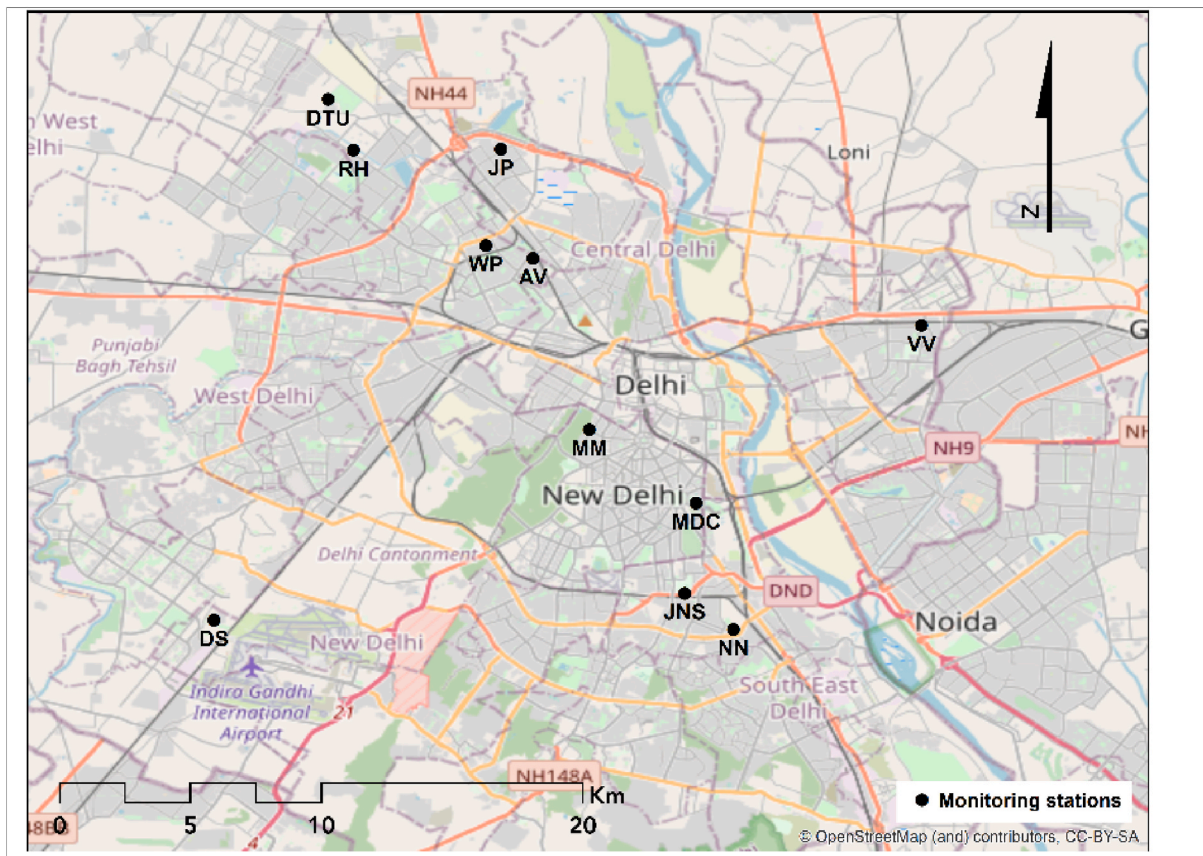


Fig. 1. (continued).

these cities and the effect of movement and business restrictions in the cities during the COVID-19 pandemic. This information is key to inform public policies to better manage traffic and air quality, and consequently public health. We analysed trends in air quality from 2017 to 2020, specifically the critical health pollutants such as particulate matter ($PM_{2.5}$ and PM_{10}), nitrogen dioxide (NO_2), ozone (O_3) and carbon monoxide (CO).

Most studies have been focused on analyzing the effect of meteorological parameters on the spread of COVID-19 (Kumar, 2020) (Srivastava, 2021), and only a few have considered the interaction of meteorology and other parameters on the reduction of air pollution concentration during the lockdown (Wu et al., 2019).

The novelty of our research lies in that it allows identifying the magnitude of reductions in activities (i.e. traffic, industrial and other) required to meet air quality standards in the cities investigated. We believe that the results of this research can help other cities in designing adaptive air quality management policies.

2. Methodology

We investigated ambient ground-level concentrations of $PM_{2.5}$, PM_{10} , NO_2 , O_3 and CO, in different geographical, and land use type areas of Mexico City, London and Delhi. Data collected during four defined different periods of lockdown restrictions were compared to the measurements for the matching period from 2017 to 2019, or 2018 to 2019 in the case of Delhi. Daily average concentrations were calculated to evaluate differences in pollutant concentrations and assess the impacts of pollution events on air quality. The percentage difference in pollutant concentrations between 2020 and the previous three years was determined for the four lockdown phases, before lockdown (BL), LD-1 the most restricted phase, LD-2, and unlock phase (UN) where restrictions were relaxed (see Table S11). Factors impacting the cities' air quality

such as meteorology, common pollution events and interventions to reduce critical health pollutants were also included in the analysis.

Fig. 1 shows the locations of selected monitoring stations in each city. Wind speed and direction was also determined for each city for the period January to August 2020 (see Figure S11). Table S12 describes the station locations in each of the three cities by geographical area and type of activity (e.g., traffic or urban background). Stations were selected to represent industrial, residential, and a mixture of residential and services land use. A minimum of 80% valid data was required for a location to be included in the data set. Missing values and untrustworthy data were excluded from the analysis.

All the measurements were taken by precision air quality (AQ) monitoring equipment with quality assurance and quality control (QA/QC) protocols for the sampling, analysis and calibration. Data was downloaded for 10 monitoring sites within Mexico City (Fig. 1) from the automatic atmospheric monitoring network (RAMA) and meteorological forecasting from the Secretariat of Environment (SEDEMA) for January 2017 to August 2020. Sites were selected in different geographical areas to represent urban traffic locations (San Agustín (SAG), Nezahualcóyotl (NEZ), Tlalnepantla (TLA), Merced (MER), Hospital General de México (HGM), UAM Xochimilco (UAX), Centro de Ciencias de la Atmósfera (CCA) and Pedregal (PED) and background locations (Tlahuac (TAH) and Cuajimalpa (CUA)). Wind speed and direction was from the Environmental Analysis Laboratory (LAA) station located north of Mexico City, 8.5 km away from the Benito Juárez International Airport. Measurements for London were downloaded for 12 locations in the Department for Environment, Food and Rural Affairs (DEFRA) Automatic Urban and Rural Network (AURN) for January 2017 to August 2020 (see Fig. 1). All sites are within the city's low emission zone (LEZ) which discourages the most polluting heavy diesel vehicles by charging an access fee. Two of the central urban background locations, Bloomsbury (CLL2) and Westminster (HORS), are also within the congestion

charge (CC) and ultra-low emissions zone (ULEZ) with stricter restrictions & charges including non-Euro 4 compliant petrol cars and non-Euro 6 compliant diesel cars. Measurements from 11 continuous ambient air quality monitoring stations covering different regions of the Delhi megacity were downloaded covering the period from 2018 to 2020. The monitoring organizations for these air quality monitoring stations include CPCB (Central Pollution Control Board) and DPCC (Delhi Pollution Control Committee). Data was downloaded from the CPCB online portal Central Control Room for Air Quality Management - Delhi NCR (CCR, 2020). CPCB provides data quality assurance or quality control (QA/QC) programs by defining rigorous protocols for the sampling, analysis and calibration.

2.1. Statistical analysis

A paired *t*-test to determine the relative effect of each lockdown phase and geographical area on the response variables was used. The

response variables were the average concentrations of PM₁₀, PM_{2.5}, CO, NO₂, and O₃. The procedure consisted of computing first the differences between average values of each response variable calculated for the periods 2017–2019 or 2018–2019 in the case of Delhi, and 2020 respectively. Then, it was tested whether this average differences between periods differed from zero. The rationale behind using the paired *t*-test lies in that we had two sets of measurements that were dependent on each other, that is, the before (years 2017–2019 or 2018–2019) and after (year 2020) average estimates for each response variable. All analyses of data from across the three cities were carried out using R (version 3.3.3; R Core Team, 2017) the packages *Openair* (version 2.5.0; Carslaw D., 2012); *reshape2* (version 1.4.3; Wickham, 2017); *dplyr* (version 7.6; Wickham, 2017) for statistical analysis and *ggplot* (version 3.2.1; Wickham, 2016) for graphics.

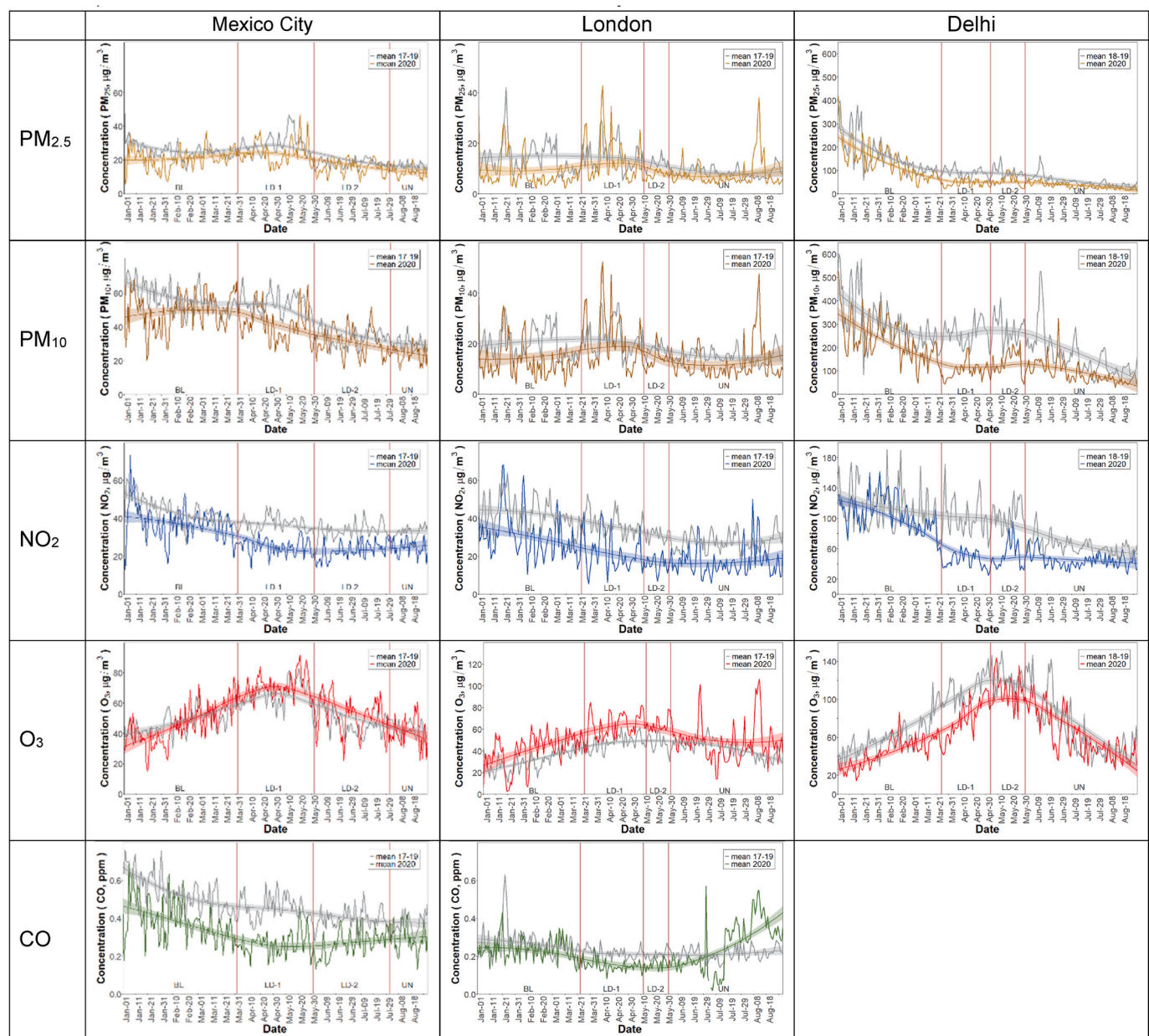


Fig. 2. Time series of PM₁₀, PM_{2.5}, NO₂, O₃ and CO concentrations during January to August 2017–2019, and different lockdown phases of COVID-19 restrictions in 2020 for Mexico City, London and Delhi.

3. Results and discussion

3.1. Daily average AQ across cities

3.1.1. Mexico city

In Mexico City concentrations of $PM_{2.5}$ and NO_2 were lower during the LD-2 and UN phases compared to BL and LD-1 for both 2017–2019 and 2020 (Fig. 2). Average concentrations of $PM_{2.5}$ were 20% lower, 23 $\mu g/m^3$ for 2017–19 and 18 $\mu g/m^3$ in 2020, whereas for PM_{10} were 16% lower, 48 $\mu g/m^3$ and 40 $\mu g/m^3$, respectively (Table 1). Average NO_2 concentrations were 30% lower, (36 $\mu g/m^3$) in 2017–19 compared to 2020. Average concentrations of CO were 39% lower, 0.38 ppm for 2017–19 compared to 0.24 ppm in 2020. O_3 concentrations showed an overall increase of 1.5%. NO_2 (38% and 36%) and CO (48% and 41%) showed higher reductions in LD-1 and LD-2 (Table 3). Which agrees with the percentage reduction in vehicular traffic with 59% and 70%, respectively. Reductions in $PM_{2.5}$ emissions were highest in BL (23%) and UN (21%), however $PM_{2.5}$ concentrations in LD-1 (17%) and LD-2 (18%) may have been impacted by high fires activity during much of May (21 May) raised average $PM_{2.5}$ concentrations up to 47 $\mu g/m^3$ (Fig. 3). PM_{10} showed a similar reduction in the BL, LD-1 and LD-2 with approximately 16% and 15% in UN. Reduction of O_3 concentrations was observed in BL with concentrations of 43 $\mu g/m^3$ (10%) and 39 $\mu g/m^3$ (13%) in UN phases, however there was an increase of 6% (70 $\mu g/m^3$) in LD-1 and 8% (52 $\mu g/m^3$) during LD-2. Average O_3 concentrations were similar with an estimate of 66 $\mu g/m^3$ in 2017–19 and 70 $\mu g/m^3$ in 2020 for LD-1. High levels of O_3 were due to the dry warm period with high solar radiation (March–May) experienced in Mexico City, a phenomenon also known as the ozone-season with atmospheric stability and clear skies which favours photochemical activity.

3.1.2. London

In London, concentrations of $PM_{2.5}$ and NO_2 were lower throughout January to August 2020 than in the same period during 2017–2019 (Fig. 2, Table 2). Average concentrations of $PM_{2.5}$ were 25% lower, 12 $\mu g/m^3$ for 2017–19 and 9 $\mu g/m^3$ in 2020, whereas for PM_{10} it was 20% lower, with 19 $\mu g/m^3$ for 2017–19 and 15 $\mu g/m^3$ in 2020. Average NO_2 concentrations were 39% lower, 35 $\mu g/m^3$ in 2017–19 and 21 $\mu g/m^3$ in 2020, (Table 3). CO concentrations showed an overall reduction of 14%, from 0.24 to 0.21 ppm. O_3 concentrations were higher across the study period with an average increase of 29%, from 42 $\mu g/m^3$ (2017–19) to 53 $\mu g/m^3$ (2020). NO_2 reductions may provide the closest comparison with traffic volumes as they demonstrate consistently strong reductions across the period investigated (29%, 40%, 49%, and 37% across the four phases investigated (see Table 3)). This result agrees with the significant decrease in NO_2 concentration due to traffic in London (Collivignarelli et al., 2021b). Reduction in $PM_{2.5}$ emissions appear to be greatest in BL (43%) then LD-2 (37%), however the reduction in $PM_{2.5}$ measurements in LD-1 (11%) may have been lessened by a southerly wind bringing particulates from across the European continent and possibly as far as the Sahara Desert (Figures S11, S12). Reductions in PM_{10} were greatest in BL (34%) followed by LD-2 (23%) and UN (18%) and similarly to the fine fraction, the LD-1 (8%) showed the lowest reduction. O_3 increased by 33%, 33%, 34% then 14% across the four phases. An uncharacteristic peak of concentrations of $PM_{2.5}$, NO_2 and O_3 was measured from August 5 to 15th 2020 (Figure S12). CO data was available for two geographical areas only (KC1, urban background and MY1, traffic, both central and within the low emission zone). These demonstrated CO reductions of 12%, 29% and 25% for the first three phases and an increase of 9% during UN. The increase appears to be measurement-instrument error from June 24th (Fig. 2 and S12). Much of the CO in London can be attributed to incomplete vehicular fuel combustion which increases during cold weather (cold running of engines). The increase in CO concentrations during the UN, when temperatures were warm, may be attributable to increased private vehicle use due to concerns regarding the use of public transport.

3.1.3. Delhi

In Delhi, the reductions in average $PM_{2.5}$ concentrations ranged from 8% to 50%, with the highest reductions observed during LD-1 from 91.6 $\mu g/m^3$ in 2018–19 to 45.9 $\mu g/m^3$ for 2020 (Table 4), whereas for PM_{10} it was 40% lower, with 247 $\mu g/m^3$ for 2018–19 and 145 $\mu g/m^3$ in 2020. Reductions in PM_{10} were greatest in LD-1 (61%) followed by UN (48%) and LD-2 (44%) (Fig. 2, Table 3). Similar reductions in average NO_2 concentrations were seen with ranges from 5 to 55%, and the greatest reductions observed during LD-1 from 100 $\mu g/m^3$ in 2018–19 to 44.5 $\mu g/m^3$ in 2020. O_3 concentrations showed an overall reduction across the period investigated with average reductions ranging from 11 to 28%, with greatest reductions during the BL phase (Table 3). This observation is contrary to those from Mexico City and London, where O_3 concentrations increased during lockdown phases. It could be speculated that NO levels in Delhi were high (higher than those observed in Mexico City and London) enough even during the lockdown to provide sufficient molecules to scavenge O_3 hence the consequent reduction in O_3 levels, as some of the NO sources in Delhi like, use of solid fuel and coal power plant are still active during the lockdown period (Manchanda et al., 2021).

In the first lockdown phase (24th January to February 23, 2020) in Wuhan, China, where the pandemic started, $PM_{2.5}$ decreased by 36.9%, NO_2 showed a decrease of approximately 53.3% and O_3 increased by 116.6% compared with the before lockdown period (1st January to January 22, 2020) (Lian et al., 2020). During the same time frame, Shi and Brasseur (2020) analysed surface measurements data and showed that 35% and 60% reduction in surface $PM_{2.5}$ and NO_2 across China. The COVID-19 restrictions also reduced urban anthropogenic emission activities across India, with different levels of the reduction attributable to different anthropogenic sources of pollutants and meteorological conditions. In the first phase of lockdown, Mahato et al. (2020) reported 60%, 39% and 53% reduction of PM_{10} , $PM_{2.5}$ and NO_2 , respectively, compared to 2019 in Delhi. The reduction of $PM_{2.5}$ concentration was 35% in Kolkata and 28% in Delhi from 22nd to March 31, 2020 (Singh and Chauhan, 2020) based on ground-level monitoring data. Using a WRF-AERMOD modelling system, Sharma et al. (2020) reported a 43% decline of $PM_{2.5}$ during March 16th to April 14th, when compared with a similar period in previous years. Dhaka et al. (2020) observed that in the first week of lockdown (25th to March 31, 2020), $PM_{2.5}$ showed large reductions (by 40–70%) compared to the pre-lockdown conditions in the Delhi-NCR region. In UK cities, the lockdown (March 23, 2020) effect on pollutants with a sharp drop in NO_2 pollution (~60% after two weeks), however, no consistent reduction was seen in $PM_{2.5}$ over the same period. Higham et al. (2020) reported that during the first 100 days of lockdown, $PM_{2.5}$ and NO_2 reduced by 26% and 36%, whereas O_3 increased by 16% compared with 2019 in London. In the Italian city Milan, $PM_{2.5}$, CO, NO_2 reduced by 47.4%, 57.6%, 61.4% respectively due to total lockdown (23rd March to 5th April) in 2020 compared to average values for 2016–2019, with O_3 increasing 253.6% during the same period (Collivignarelli et al., 2020). Surface measurements for the Barcelona region (Spain), revealed a decrease in NO_2 of 47%–51.4%, and increase in O_3 of 33.0%–57.7% during lockdown (14th - March 30, 2020) compared to before lockdown (16th February to March 13, 2020) (Tobías et al., 2020). In comparison with many cities, Tokyo, Japan, had lower reductions of $PM_{2.5}$ (24.1%), NO_2 (25.8%), CO (3.1%) and O_3 (5.3%) during its COVID-19 restrictions (7th April to May 24, 2020) compared with average concentrations in 2017–2019 (Fu et al., 2020).

3.2. Spatial (geographic) variations in AQ changes

3.2.1. Mexico city

We split each city into discrete geographical areas to determine any difference in the impacts of COVID-19 restrictions according to location (Figs. 1 and 2 and S12, Table S12). Air quality monitoring stations in Mexico City were distributed in five geographical areas with different land use, northeast and northwest industrial with old and new gasoline

Table 1

Average pollutant concentrations and percentage change by lockdown phase and geographical area in Mexico City during 2017–2019 and 2020. Statistical significance values are shown with letters.

Pollutant	Phase	NorthWest (NW)				NorthEast (NE)				Center (CE)				SouthWest (SW)				SouthEast (SE)			
		2017–2019	2020	%	p	2017–2019	2020	%	p	2017–2019	2020	%	p	2017–2019	2020	%	p	2017–2019	2020	%	p
PM _{2.5} (µg/m ³)	BL	26.5 ± 7.6	22.7 ± 6.6	−14.7	a	28.8 ± 10.6	24.8 ± 7.6	−13.8	b	27.7 ± 9.2	21.9 ± 8.5	−21.0	a	21.0 ± 7.2	17.1 ± 6.7	−18.2	a	25.5 ± 9.0	19.4 ± 6.4	−23.9	a
	LD-1	30.1 ± 9.2	23.7 ± 6.3	−10.4	a	31.4 ± 12.8	27.6 ± 9.0	−12.0	c	30.5 ± 11.1	25.5 ± 7.4	−16.5	b	27.4 ± 11.6	22.6 ± 8.4	−17.3	b	28.2 ± 13.6	23.7 ± 8.1	−15.7	c
	LD-2	21.1 ± 5.3	19.9 ± 5.4	−5.6	ns	18.3 ± 6.4	16.6 ± 5.7	−9.2	ns	19.5 ± 6.0	16.9 ± 5.6	−13.3	c	15.2 ± 4.2	13.6 ± 4.6	−10.6	c	17.2 ± 4.7	13.7 ± 4.6	−20.1	a
	UN	19.5 ± 6.1	15.0 ± 3.5	−3.5	c	14.6 ± 4.6	13.9 ± 4.5	−4.8	ns	17.5 ± 4.9	15.5 ± 5.2	−11.3	ns	15.2 ± 4.0	12.5 ± 3.9	−17.5	c	14.5 ± 3.5	12.0 ± 3.5	−17.0	c
PM ₁₀ (µg/m ³)	BL	56.7 ± 15.1	49.3 ± 13.4	−13.1	a	70.6 ± 24.2	61.5 ± 18.7	−12.9	b	51.8 ± 13.0	44.5 ± 11.7	−14.1	a	40.2 ± 11.0	33.2 ± 12.2	−17.5	a				
	LD-1	55.9 ± 14.8	39.8 ± 11.4	−28.8	a	59.3 ± 19.0	51.2 ± 14.4	−13.6	c	45.5 ± 11.5	38.2 ± 10.8	−15.9	c	44.0 ± 13.6	33.4 ± 10.3	−24.0	a				
	LD-2	37.4 ± 10.4	32.5 ± 9.2	−13.1	a	35.1 ± 12.1	36.5 ± 11.3	4.1	ns	29.6 ± 5.9	23.3 ± 7.6	−21.1	a	27.9 ± 8.4	23.2 ± 4.8	−17.1	a				
	UN	32.7 ± 8.7	25.7 ± 6.8	−21.4	a	30.4 ± 8.3	27.3 ± 7.2	−10.2	d	25.7 ± 5.9	20.4 ± 5.7	−20.5	b	26.6 ± 6.0	18.7 ± 4.8	−29.7	a				
NO ₂ (µg/m ³)	BL	51.6 ± 12.9	45.5 ± 12.1	−11.7	b	40.4 ± 9.9	35.7 ± 8.9	−11.7	a	52.7 ± 13.3	44.8 ± 11.0	−14.9	a	36.0 ± 11.2	30.0 ± 10.0	−16.7	a	36.0 ± 10.2	30.2 ± 8.9	−16.2	c
	LD-1	46.4 ± 8.8	34.0 ± 6.2	−26.7	a	35.2 ± 8.0	25.7 ± 5.3	−26.9	a	45.2 ± 10.1	30.5 ± 6.7	−32.5	a	30.1 ± 7.5	16.9 ± 5.0	−43.9	a	29.8 ± 7.0	21.3 ± 4.3	−28.7	a
	LD-2	39.7 ± 9.8	30.9 ± 6.5	−22.0	a	30.4 ± 8.3	21.5 ± 5.5	−29.3	a	39.9 ± 9.8	27.4 ± 6.3	−31.4	a	29.1 ± 7.0	17.0 ± 4.2	−41.7	a	28.1 ± 6.8	19.5 ± 4.7	−30.6	a
	UN	40.8 ± 9.4	31.4 ± 5.0	−22.9	a	29.8 ± 7.1	22.8 ± 5.9	−23.6	a	40.4 ± 9.2	31.2 ± 6.9	−22.8	a	26.8 ± 6.0	19.6 ± 4.0	−26.9	a	26.0 ± 6.4	22.4 ± 5.2	−14.0	c
O ₃ (µg/m ³)	BL	37.9 ± 11.3	36.2 ± 12.4	−4.6	ns	42.6 ± 12.4	40.2 ± 10.8	−5.6	c	45.3 ± 13.9	42.7 ± 13.8	−5.7	d	51.8 ± 15.1	51.4 ± 16.8	−0.8	ns	52.7 ± 13.6	52.1 ± 14.9	−1.2	ns
	LD-1	54.9 ± 11.6	58.6 ± 7.8	6.8	b	62.8 ± 14.0	62.4 ± 10.2	−0.6	ns	64.4 ± 15.1	73.2 ± 10.1	13.7	a	70.0 ± 17.2	79.9 ± 11.8	14.1	a	75.2 ± 16.5	74.6 ± 9.4	−0.8	ns
	LD-2	39.8 ± 12.5	40.5 ± 9.2	1.7	ns	43.3 ± 13.6	46.6 ± 10.9	7.6	ns	44.7 ± 15.4	52.3 ± 13.9	17.0	b	51.6 ± 15.9	59.7 ± 14.6	15.8	b	51.7 ± 14.7	56.5 ± 14.5	9.1	d
	UN	39.4 ± 8.8	28.0 ± 8.8	−29.0	a	39.2 ± 10	36.4 ± 8.2	−7.0	ns	41.8 ± 10.5	37.4 ± 10.4	−10.6	d	45.7 ± 12.9	48.5 ± 12.7	6.1	ns	48.4 ± 11.8	45.2 ± 11.2	−6.6	ns
CO (ppm)	BL	0.61 ± 0.25	0.44 ± 0.16	−27.8	a	0.62 ± 0.20	0.48 ± 0.16	−22.9	a	0.63 ± 0.28	0.43 ± 0.16	−30.9	a	0.42 ± 0.18	0.28 ± 0.11	−33.4	a	0.42 ± 0.14	0.36 ± 0.13	−13.9	c
	LD-1	0.48 ± 0.16	0.29 ± 0.08	−39.8	a	0.52 ± 0.22	0.33 ± 0.10	−37.3	a	0.53 ± 0.23	0.30 ± 0.10	−43.5	a	0.38 ± 0.16	0.20 ± 0.07	−46.9	a	0.36 ± 0.12	0.24 ± 0.06	−34.0	a
	LD-2	0.50 ± 0.19	0.30 ± 0.09	−38.7	a	0.44 ± 0.18	0.29 ± 0.09	−34.0	a	0.43 ± 0.19	0.31 ± 0.10	−28.0	a	0.33 ± 0.13	0.19 ± 0.06	−42.7	a	0.34 ± 0.15	0.24 ± 0.07	−30.7	a
	UN	0.44 ± 0.16	0.36 ± 0.09	−17.3	b	0.46 ± 0.14	0.31 ± 0.12	−33.2	a	0.41 ± 0.15	0.37 ± 0.11	−8.5	ns	0.27 ± 0.11	0.22 ± 0.07	−19.7	b	0.34 ± 0.11	0.29 ± 0.09	−15.3	d

BL-phase (1st January – 31st March); LD-phase 1 (1st April – 31st May); LD-phase 2 (1st June – 31st July); UN-phase (1st August – 31st August).

a: $p \leq 0.0001$; b: $p \leq 0.001$; c: $p \leq 0.01$; d: $p \leq 0.05$; ns: $p > 0.05$.

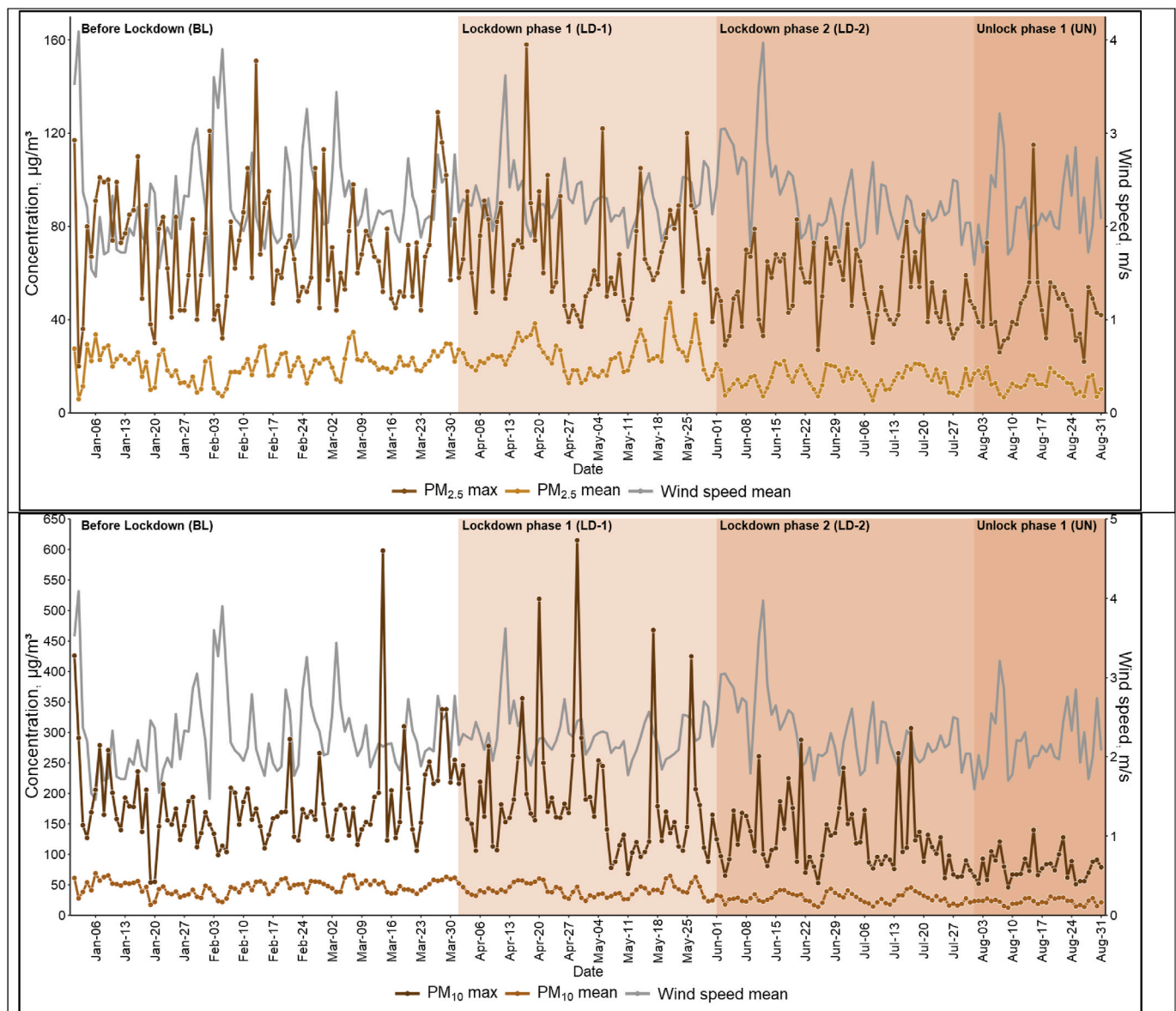


Fig. 3. Times series of 24-h average and hourly maximum concentrations of $PM_{2.5}$, PM_{10} and wind speed during different phases of COVID-19 restrictions in Mexico City.

and diesel vehicles, central with a mixture of offices and with light-duty vehicles and modern heavy-duty diesel buses, southeast and southwest residential neighborhood with light-duty vehicles (Table S12). Significant percentage decrease of CO and NO_2 were observed in all geographical areas, with up to 47.1% and 43.9% respectively, during LD-1 and LD-2, with the largest in the SW and central areas where the highest reduction in traffic was registered (>78%). In contrast, O_3 showed a maximum increase of 14.1 (SW) and 17% (C) in phase 1 and 2, with concentrations > 70 $\mu g/m^3$ of O_3 (Table 1). This was due to prevalent winds coming from Sierras de las Cruces and Ajusco-Chichinautzin mountains located in the south of the city. Greater decreases of $PM_{2.5}$ were observed in south and central areas during BL ranging from 18 to 24% and lower at the northern areas with ~14%. The highest $PM_{2.5}$ concentrations during 2020 were measured at NE with 25 $\mu g/m^3$ whereas for 2017–2019 with 29 $\mu g/m^3$.

3.2.2. London

London's air quality monitoring locations were split into four geographical zones, central, north, west and southeast (SE). Locations in

the central zone would usually have dense vehicle traffic, the west zone included Heathrow Airport, motorway and industrial emissions, the north locations were urban background, and SE locations included suburban background (Table S12). For LD-1 and after, the greatest reduction in NO_2 and $PM_{2.5}$ concentrations were measured in the west, followed by central then north zones, suggesting that reductions in motorway use, airport industrial activity and retail had the greatest impact on air quality improvement. The greatest O_3 increases from LD-1 were identified in the central and west zones. The suburban SE zones had the smallest changes in air quality compared with previous years, which may indicate a higher proportion of domestic rather than business traffic.

3.2.3. Delhi

In Delhi, air quality monitoring stations are located in the mixed region of commercial-residential, industrial-residential and industrial location in urban and suburban regions. In Delhi, NO_2 average concentrations were greatest in the north of the city, whereas the lowest values occurred in the NE area of the city (Table 4 and Graphical

Table 2

Average pollutant concentrations and percentage change by lockdown phase and geographical area in London during 2017–2019 and 2020. Statistical significance values are shown with letters.

Pollutant	Phase	North				Central				West				Southeast			
		2017–2019	2020	%	p	2017–2019	2020	%	p	2017–2019	2020	%	p	2017–2019	2020	%	p
PM_{2.5} ($\mu\text{g}/\text{m}^3$)	BL	16.9 \pm 13.6			ns	14.9 \pm 12.2	8.5 \pm 6.2	–43	a	12.6 \pm 11.3	7.5 \pm 6.0	–41	a	13.3 \pm 11.0	7.4 \pm 6.0	–44	a
	LD-1	14.2 \pm 8.5			ns	15.9 \pm 10.7	13.9 \pm 9.0	–12	ns	14.4 \pm 10.3	13.1 \pm 8.7	–9	ns	16.3 \pm 11.8	14.4 \pm 9.3	–12	ns
	LD-2					12.4 \pm 6.5	7.9 \pm 2.7	–37	a	11.1 \pm 6.4	6.1 \pm 1.6	–45	a	11.9 \pm 6.9	8.0 \pm 2.4	–33	a
	UN					8.9 \pm 4.7	8.1 \pm 5.5	–10	ns	7.5 \pm 3.8	5.7 \pm 4.0	–25	d	8.3 \pm 4.2	8.6 \pm 6.8	4	ns
PM₁₀ ($\mu\text{g}/\text{m}^3$)	BL					21.5 \pm 13.0	14.6 \pm 7.6	–32	a	16.7 \pm 13.0	11.5 \pm 7.0	–31	a	16.3 \pm 11.1	12.0 \pm 6.9	–26	c
	LD-1					23.1 \pm 12.4	21.1 \pm 10.8	–9	ns	20.6 \pm 11.8	21.0 \pm 11.4	2	ns	25.3 \pm 16.5	20.6 \pm 11.4	–19	d
	LD-2					18.9 \pm 8.1	13.2 \pm 3.4	–30	a	16.0 \pm 7.6	15.1 \pm 5.9	–6	ns	12.6 \pm 5.6	11.8 \pm 3.1	–6	ns
	UN					15.8 \pm 6.7	12.5 \pm 7.0	–21	b	13.3 \pm 35.4	11.7 \pm 6.5	–12	ns	11.5 \pm 5.7	12.0 \pm 8.6	5	ns
NO₂ ($\mu\text{g}/\text{m}^3$)	BL	30.3 \pm 14.3	19.3 \pm 12.3	–36	a	51.7 \pm 26.3	37.2 \pm 20.6	–28	a	47.8 \pm 22.2	32.7 \pm 14.5	–32	a	25.8 \pm 13.5	17.6 \pm 12.7	–32	a
	LD-1	22.5 \pm 8.4	14.5 \pm 7.6	–35	a	47.0 \pm 26.3	26.7 \pm 14.7	–43	a	40.8 \pm 19.6	21.3 \pm 11.1	–48	a	22.1 \pm 10.8	17.6 \pm 12.2	–20	d
	LD-2	17.4 \pm 6.4	8.7 \pm 3.4	–50	a	42.0 \pm 25.6	20.8 \pm 10.3	–50	a	33.3 \pm 17.0	14.4 \pm 6.4	–57	a	17.3 \pm 7.2	11.3 \pm 5.6	–35	b
	UN	15.2 \pm 6.6	10.0 \pm 6.0	–34	a	36.2 \pm 27.5	21.5 \pm 12.8	–41	a	31.1 \pm 16.9	16.2 \pm 10.0	–48	a	13.6 \pm 7.0	13.6 \pm 8.7	0	ns
O₃ ($\mu\text{g}/\text{m}^3$)	BL	39.1 \pm 19.4	50.6 \pm 18.9	29	a	26.7 \pm 17.8	37.6 \pm 19.7	41	a	29.8 \pm 17.9	39.6 \pm 17.2	33	a	38.2 \pm 18.8	51.1 \pm 19.0	34	a
	LD-1	57.5 \pm 13.0	68.5 \pm 12.1	19	a	44.1 \pm 18.3	64.4 \pm 15.1	46	a	45.0 \pm 14.5	59.7 \pm 12.5	33	a	53.8 \pm 14.4	63.0 \pm 15.0	17	ns
	LD-2	56.4 \pm 13.9	72.0 \pm 6.5	28	a	44.2 \pm 18.4	62.4 \pm 10.5	41	a	44.6 \pm 16.9	61.6 \pm 8.2	38	a	53.8 \pm 13.8	–		ns
	UN	51.2 \pm 15.3	55.1 \pm 20.5	8	ns	39.4 \pm 18.5	48.1 \pm 22.2	22	b	38.7 \pm 15.1	45.6 \pm 18.2	18	b	50.0 \pm 15.3	48.0 \pm 21.2	–4	ns
CO (ppm)	BL					0.27 \pm 0.18	0.23 \pm 0.18	–12	c								
	LD-1					0.21 \pm 0.10	0.12 \pm 0.15	–24	a								
	LD-2					0.21 \pm 0.10	0.23 \pm 0.13	9	c								
	UN					0.21 \pm 0.10	0.23 \pm 0.13										
						0.21 \pm 0.10	0.23 \pm 0.13										
						0.21 \pm 0.10	0.23 \pm 0.13										

BL-phase (1st January–22nd March); LD-phase 1 (23rd March to 11th May); LD-phase 2 (12th May to 31st May); UN-phase (1st June to 31st August).

a: $p \leq 0.0001$; b: $p \leq 0.001$; c: $p \leq 0.01$; d: $p \leq 0.05$; ns: $p > 0.05$.

Abstract). Regarding O₃ average concentrations, our analyses revealed the greatest and lowest mean values in the SW and NW areas of Delhi, respectively. Greater PM_{2.5} average concentrations were observed in NW Delhi whereas the lowest average concentrations occurred in northern Delhi.

3.3. Land use type variations in AQ changes

3.3.1. Mexico city

We investigated variations in air quality related to land-use type at monitoring locations, such as traffic, industrial or background as categorised by the monitoring authorities (Tables S13 and S14). In the case of Mexico City, only two types of monitoring locations were available, i.

e., background and traffic. Similar percentage reductions in air pollutants were observed for both these land uses. Background CO and NO₂ percentage decrease were up to 40% and 44%, respectively, whereas for traffic stations it was up to 42% and 32%, for CO and NO₂, respectively. PM_{2.5} showed a 20% reduction in traffic locations. O₃ concentrations were higher in background locations than traffic locations during the lockdown phases. This is as expected because fresh NO from traffic will scavenge O₃. In Mexico City O₃ production is generally volatile organic compounds (VOC) limited (Jaimes-Palomera et al., 2016; Peralta et al., 2020) which has been reinforced by the weekend effect showing large reduction in NO_x and CO but not in O₃ compared to weekday concentrations (Stephens et al., 2008).

Table 3
Inter-city comparison of average pollutant concentration changes and percentage changes during four phases of COVID-19 restrictions in 2020.

Pollutant	BL			LD-1			LD-2			UL			Average	
	2017–2019	2020	Change (%)	2017–2019	2020	Change (%)	2017–2019	2020	Change (%)	2017–2019	2020	Change (%)	2017–2019	2020
Mexico City	PM _{2.5} (µg/m ³)	26.1	20.1	-23.1	29.5	24.6	-16.6	18.2	14.9	16.4	13	-20.6	23	18
	PM ₁₀ (µg/m ³)	55.1	45.9	-16.8	52.4	44.0	-16.1	51.4	42.9	31.1	26.6	-14.5	48	40
	NO ₂ (µg/m ³)	43	35.2	-18.1	36.9	23	-37.7	32.7	20.9	31.5	22.9	-27.3	36	26
	O ₃ (µg/m ³)	47.3	42.8	-9.6	65.8	69.6	5.7	47.9	51.5	45.1	39.2	-13	52	51
	CO (ppm)	0.478	0.315	-34.1	0.406	0.209	-48.4	0.335	0.197	0.319	0.221	-30.6	0.38	0.24
London	PM _{2.5} (µg/m ³)	13.9	7.9	-43.2	15.6	13.9	-10.9	11.9	7.5	8.4	7.7	-8.2	12	9
	PM ₁₀ (µg/m ³)	19.9	13.1	-34.0	22.7	21.0	-7.5	17.7	13.6	14.8	12.2	-17.8	19	15
	NO ₂ (µg/m ³)	42.5	30.1	-29.1	37	22.1	-40.2	31.4	15.9	27.6	17.3	-37.4	35	21
	O ₃ (µg/m ³)	31.1	41.4	33.2	47.8	63.5	33	47.5	63.8	42.3	48.4	14.3	42	54
	CO (ppm)	0.285	0.287	-12.2	0.223	0.159	-28.9	0.218	0.165	0.224	0.244	9.2	0.24	0.21
Delhi ^a	PM _{2.5} (µg/m ³)	152.8	140.5	-8	91.6	45.9	-49.8	89	58.7	50.3	33.6	-33.2	96	70
	PM ₁₀ (µg/m ³)	290.6	239.3	-17.6	263.4	102.7	-61.0	268.2	151.3	164.2	85.89	-47.7	247	145
	NO ₂ (µg/m ³)	114.1	108.1	-5.3	100	44.5	-55.5	105.5	56.9	62.9	44.4	-29.4	96	63
	O ₃ (µg/m ³)	61.1	43.9	-28	110.7	84.7	-23.5	120.3	107.3	68.9	59.4	-13.8	90	74

^a 2018–2019 averages.

3.3.2. London

London data used in this investigation included urban and suburban background, urban traffic, and urban industrial location types. Vehicle traffic is recognised as the major source of air pollution in London (Greater London Authority, 2020). In keeping with this, the greatest improvements in air quality were seen at the urban traffic location, with reductions of 67% (76–23 µg/m³) for NO₂, 63% (16–6 µg/m³) for PM_{2.5}, 33% for CO, and an increase of 88% (30–56 µg/m³) in O₃ (see Table SI3). Vehicle miles travelled in London reduced by as much as 63% during LD-1 (range 53%–63%), 39%–43% in LD-2, 3%–20% in BL (2nd to March 22, 2020) and 38 to 27% in UN (1st to June 22, 2020) (Pishue & Markezich, 2020). Urban background, suburban background and urban industrial locations all demonstrated similar reductions in PM_{2.5} measurements (~30%–44% over BL and LD-2). NO₂ reductions were next greatest at the industrial location, which is also close to Heathrow Airport. O₃ concentrations increased across the city, as might be expected, with warm weather and less NO being produced by vehicles. These increases were highest at the urban traffic location (88%) but still significant at other locations (15%–30%) which agreed with the results reported in most European cities where O₃ concentration increased during the COVID-19 lockdown period (Collivignarelli et al., 2021b). CO reduction in the urban background location was not as great as at the urban traffic location but still a 20% improvement. During UN an unexpectedly large increase in CO at KC1 (70%) was observed, possibly an instrument measurement error (Fig. 2 and SI2).

3.3.3. Delhi

In Delhi, the change of pollutant's concentration level was different given the land use type. The reduction of PM_{2.5} was highest for commercial-residential in LD-1 (52%) and LD-2 (37%), compared to industrial-residential (LD-1 45%; LD-2 30%) and industrial (LD-1 52% and LD-2 33%) (Table SI4). This was mainly because the small food vendors in residential areas used coal for cooking purposes, which was restricted due to lockdown (Chitlangia, 2015).

3.4. Inter-city comparison

Pollutant profiles for each city (Fig. 2 and SI2) illustrate differences in pollutant concentrations to which citizens are exposed to, the impacts of restrictions imposed during lockdown phases, as well as impacts of meteorology and other phenomena such as forest fires and crop stubble burning.

PM_{2.5} average concentrations were highest in Delhi (96 µg/m³ in January to August 2018–2019 and 70 µg/m³ in 2020 with a net reduction of 31%), then Mexico City (23 µg/m³ in January to August 2017–2019 and 18 µg/m³ in 2020 with a net reduction of 20%), and London (12 µg/m³ and 9 µg/m³ respectively with a net reduction of 25%). Fig. 2 and SI3 show that PM_{2.5} concentrations in Delhi were much higher in the cold winter months at the beginning of the year. Wind speeds in Delhi were consistently low throughout the year (Figure SI1) not rising above 2 m/s, whereas London citizens benefit from the cleansing effect of winds often over 6 m/s especially in the colder months when PM_{2.5}, NO₂ and CO would otherwise be much higher. Wind speeds in Mexico City fall in between Delhi and London. PM_{2.5} concentrations in each of the cities investigated can be strongly influenced by long-range transport of particulates. Forest fires and seasonal crop burning impact the air in Mexico City from January to March, regional crop stubble burning impacts Delhi's citizens in October, and London's citizens receive particulates from across Europe and the Sahara when prevailing winds are from the south and east (Figs. 2 and 3, Graphical Abstract and SI3). Fig. 3, Graphical Abstract and SI3 demonstrate that the maximum PM_{2.5} measurements in Mexico City were much greater than the average concentrations. In Delhi and London maximum and average concentrations of PM_{2.5} were much closer to each other. The greatest difference in NO₂ concentrations for the same time phase in previous years and 2020 was observed in the LD-1. For

Table 4

Average pollutant concentrations and percentage change by lockdown phase and geographical area in Delhi during 2018–2019 and 2020. Statistical significance values are shown with letters.

Pollutant	Phase	North (N)				NorthWest (NW)				NorthEast (NE)				Center (CE)				SouthWest (SW)			
		2018–2019	2020	%	p	2018–2019	2020	%	p	2018–2019	2020	%	p	2018–2019	2020	%	p	2018–2019	2020	%	p
PM_{2.5} (µg/m³)	BL	114.3 ± 61.4	113.0 ± 63.3	−1	d	164.9 ± 86.8	147.5 ± 74.8	−11	a	144.1 ± 78.0	135.4 ± 68.5	−6	c	150.0 ± 86.0	136.7 ± 84.0	−9	b	135.2 ± 67.6	135.8 ± 75.6	0	ns
	LD-1	76.7 ± 29.6	36.6 ± 14.0	−52	a	101.8 ± 34.3	51.0 ± 17.3	−50	a	89.9 ± 34.5	42.2 ± 18.5	−53	a	76.3 ± 27.7	40.5 ± 15.3	−47	a	86.9 ± 30.9	46.3 ± 17.6	−47	a
	LD-2	87.8 ± 33.7	42.9 ± 18.3	−51	a	103.1 ± 338.2	63.1 ± 29.9	−39	a	89.7 ± 32.4	54.2 ± 22.9	−40	a	79.8 ± 28.9	46.3 ± 18.7	−42	a	89.4 ± 33.8	52.3 ± 24.3	−42	a
	UN	49.6 ± 41.9	27.5 ± 12.6	−45	a	54.9 ± 37.5	37.0 ± 14.8	−33	a	48.3 ± 27.9	38.1 ± 15.5	−21	a	44.1 ± 30.6	29.8 ± 13.8	−32	a	46.7 ± 22.8	31.5 ± 12.6	−33	a
PM₁₀ (µg/m³)	BL	251.0 ± 104.2	191.5 ± 83.5	−24	a	306.3 ± 123.5	243.5 ± 98.9	−21	a	±114.0	223.8 ± 85.2	−16	a	256.2 ± 111.2	220.4 ± 99.3	−14	a	327.7 ± 118.6	296.6 ± 109.8	−9	c
	LD-1	231.7 ± 89.2	80.8 ± 30.1	−65	a	288.3 ± 102.5	109.3 ± 40.6	−62	a	252.7 ± 75.2	99.3 ± 35.7	−61	a	222.8 ± 75.0	91.9 ± 34.1	−59	a	261.9 ± 91.0	107.4 ± 41.9	−59	a
	LD-2	253.5 ± 87.9	113.7 ± 42.1	−55	a	299.9 ± 97.6	146.9 ± 63.3	−51	a	262.1 ± 85.3	138.0 ± 45.1	−47	a	235.5 ± 84.1	127.6 ± 48.3	−46	a	305.5 ± 100.0	154.4 ± 58.7	−49	a
	UN	137.1 ± 110.7	62.4 ± 36.0	−55	a	173.5 ± 130.0	93.5 ± 44.1	−46	a	146.4 ± 95.0	96.7 ± 43.1	−34	a	142.0 ± 122.0	75.4 ± 46.9	−47	a	216.5 ± 129.9	100.7 ± 59.4	−54	a
NO₂ (µg/m³)	BL	131.2 ± 37.2	104.4 ± 28.9	−20	a	116.7 ± 55.4	114.4 ± 92.4	−2	ns	72.9 ± 31.3	91.3 ± 36.8	25	d	129.7 ± 75.7	101.1 ± 39.3	−22	a	84.8 ± 51.8	107.3 ± 25.1	26	b
	LD-1	126.8 ± 35.3	30.2 ± 8.5	−76	a	99.6 ± 50.7	57.9 ± 45.2	−42	a	65.5 ± 32.0	48.0 ± 12.9	−27	a	117.9 ± 48.2	30.0 ± 11.7	−75	a	82.9 ± 42.0	32.5 ± 9.3	−61	a
	LD-2	143.4 ± 40.7	46.7 ± 20.6	−67	a	104.9 ± 50.8	52.8 ± 25.7	−50	a	71.2 ± 28.5	68.3 ± 27.7	−4	ns	114.2 ± 44.9	47.9 ± 23.2	−58	a	78.2 ± 26.2	46.2 ± 17.5	−41	a
	UN	79.6 ± 27.4	39.7 ± 11.9	−50	a	60.1 ± 29.6	40.7 ± 13.6	−32	a	43.0 ± 18.6	46.8 ± 20.8	9	ns	73.6 ± 25.4	52.7 ± 13.4	−28	a	55.3 ± 18.4	50.2 ± 12.6	−9	c
O₃ (µg/m³)	BL	51.1 ± 25.1	35.2 ± 11.4	−31	a	60.5 ± 29.0	27.3 ± 19.5	−55	a	67.9 ± 26.2	54.4 ± 21.8	−20	b	65.2 ± 28.7	69.9 ± 29.9	7	c	70.3 ± 31.7	60.3 ± 23.7	−14	ns
	LD-1	96.2 ± 17.5	47.5 ± 11.4	−51	a	101.9 ± 37.6	44.8 ± 31.2	−56	a	123.0 ± 30.6	101.9 ± 27.2	−17	a	118.8 ± 29.9	137.3 ± 41.0	16	c	120.5 ± 22.9	122.5 ± 31.6	2	ns
	LD-2	112.3 ± 28.9	98.6 ± 75.5	−13	ns	112.6 ± 32.4	74.2 ± 36.5	−34	a	122.2 ± 43.4	103.6 ± 32.5	−15	d	131.3 ± 31.7	168.2 ± 39.8	28	a	135.5 ± 34.3	152.8 ± 31.1	13	d
	UN	66.6 ± 43.2	67.1 ± 46.0	1	b	61.1 ± 37.6	48.9 ± 43.5	−20	b	81.9 ± 49.5	67.4 ± 24.1	−18	b	77.0 ± 45.7	63.7 ± 44.4	−17	b	73.7 ± 46.9	79.4 ± 41.5	8	ns

BL-phase (1st January –22nd March); LD-phase 1 (23rd March to 11th May); LD-phase 2 (12^h May to 31st May); UN-phase (1st June to 31st August).a: $p \leq 0.0001$; b: $p \leq 0.001$; c: $p \leq 0.01$; d: $p \leq 0.05$; ns: $p > 0.05$.

example, in Delhi city-wide average concentrations were reduced from 100 to 45 $\mu\text{g}/\text{m}^3$ (56% reduction) (see Table 3). Similarly, calculated NO_2 average changes during the LD-1 were 37 to 22 $\mu\text{g}/\text{m}^3$ (40% reduction) in London and 37 to 23 (38% reduction) in Mexico City. NO_2 reductions remained high during the LD-2 across the three cities, 46% in Delhi, 49% in London and 36% in Mexico City. NO_2 concentrations in all the cities are commonly associated with traffic, in particular older diesel engines. Similar NO_2 reductions ranging from 27% to 63% have been reported in Asia's urban centers (Kanniah et al., 2020; Suhaimi et al., 2020).

Maji et al. (2021) reported that there are no systematic correlations were observed between pollutants and weather parameters from March to June in Delhi city. This was due to the interplay between two different weather and air pollution emission scenarios from March to June. The end of March corresponds to the LD-1, where all the services were closed and there were no active emission sources. However, June represents the UN, where essential services were opened gradually. These scenarios lead to different emission patterns, which led to the inconsistencies observed in the correlation coefficients. Worthy of note is the negative correlation coefficient between relative humidity and air pollutants, and the positive correlation coefficient between temperature and air pollution is increased in 2020, compared to 2018–2019. In London, there are no unusual meteorological conditions in the past three years (i.e., temperature and relative humidity) (Jephcote et al., 2021). In the UK air quality has been negatively influenced by a significant change in meteorology between the weeks preceding and following the lockdown in addition to changes (both positive and negative) arising from actions in response to COVID-19. And the meteorological conditions have led to higher $\text{PM}_{2.5}$ during lockdown than the average experienced in equivalent calendar periods from previous years in some parts of the UK (Jephcote et al., 2021).

Several methodological uncertainties are involved in our approach which may limit to some degree its applicability to quantifying air pollution anomalies. First, for the January to August differential, we calculated average pollutant values as the difference between 2020 and the average of a 3-y baseline (2017–2019) and 2-y in the case of Delhi. Previous studies suggest using more than 5-year historical data to estimate trends in pollutants concentrations (Zangari et al., 2020). However, lack of data availability and changing AQ policies implemented historically in Mexico City, restricted the time frame analysed. Second, the start and duration of lockdown restrictions among the three cities were different, so comparisons in air pollution reductions could be slightly biased. Third, we acknowledge that using chemical transport models with high coverage area could have helped us to better understand the influence of meteorology in pollutants reduction, thus allowing quantification of the transboundary air pollutants. Fourth, machine learning algorithms trained to account for meteorological influence in air quality time series could have been used such as those developed by Keller et al. (2021), Peletin et al. (2021) and Ryan et al. (2021). However, this analysis was outside the scope of this paper. Fourth, we recommend further kinetic studies to understand the optimized reduction of pollutants concentration to minimize O_3 increment (Chen et al., 2021).

4. Conclusions

The COVID-19 pandemic has reduced human activity, causing a slowdown in economic activities and transportation, and the reduction of most ground-level air pollutants in urban cities. In Mexico City, London and Delhi air quality changes following the lockdown are evident from ground measurements at different monitoring stations, most notably, a drop in $\text{PM}_{2.5}$, CO and NO_2 concentrations. Air pollutants did not respond equally in the three cities under this study during the period evaluated. Pollutants dominated by transportation emissions showed clear reductions in CO and NO_2 . Fine particles showed a different pattern among cities. In Mexico City an increase in $\text{PM}_{2.5}$ was

observed due to several forest fires during the tightest restriction phase. NO_2 concentration dropped quickly when lockdown restrictions were imposed in all three cities, as traffic flow reduced. This study has demonstrated differences in O_3 production across the cities, being NO_x limited in Delhi, and VOC limited in London and Mexico City. The O_3 changed should be further investigated.

We found major reductions in $\text{PM}_{2.5}$, PM_{10} , NO_2 and CO across the three cities for the lockdown phases and increases in O_3 in London and Mexico City but not in Delhi. We discussed the city-wide changes in air quality in these cities during the four selected phases of restrictions and looked thoroughly at changes related to geography and land use types. Our study demonstrates the extent of air quality improvements that are possible with reductions in vehicle traffic. This was possible to quantify due to the strict restrictions in the use of private cars during the lockdowns, resulting in the decrease of NO_2 and CO in the urban environment. Our results showed that to reduce O_3 and PM control strategies in road traffic in addition to the regulation of VOC from the industry and the control of forest fires should be considered.

PM reductions in London and Mexico were partially significant. This was due to meteorological effects, particularly in London, and local forest fires close to Mexico City. The greatest reduction in air pollutant concentrations was found in Delhi. This was also the most clearly evidenced improvement since there were limited meteorological effects. It is well documented that the sectors contributing more to pollutant emissions including traffic mobility, industry, energy productions and agriculture, vary across cities. Therefore, better technologies associated with public transport and cleaner energies should be reinforced. Limiting vehicle use is not enough to transform these cities AQ to improve citizens health – additional interventions may be required, such as changing vehicle type.

The restrictions placed on transport, industry and construction activities due to COVID-19 lockdown have significantly improved air quality throughout the world. This viral scenario has shown that shutting down a city can significantly improve atmospheric pollution loads in urban areas and avoid premature deaths due to air pollution. This is however, an unrealistic solution to the problem since there might be untested economic drawbacks. Past studies reported that reduction in air pollution due to COVID-19 lockdown was the reason for avoided excess deaths of 49,900 around the world, but in some countries like in the UK, the O_3 -attributed premature death increased due to an increase of O_3 concentration during the lockdown (Venter et al., 2021). Giani et al. (2020) estimated that 24,200 and 2190 premature deaths were averted in China and Europe, respectively due to a decrease in $\text{PM}_{2.5}$ during the lockdown. The lowest levels of pollution concentration due to the COVID-19 restriction can be considered as baseline threshold values of air pollutants. These baseline levels can provide a steady response to the overall air quality of a city and it may inform policymakers to set new target limits to fine-tune air quality standards and develop strategies for possible attenuation of atmospheric pollution, keeping in mind that the solutions must be realistic from an economic/development sense and in the context of managing the people's comfort.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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Credit author statement

E. Vega: Writing – original draft, Formal analysis, Conceptualization, Writing – review & editing, Supervision, Funding acquisition, Mexican project administration. A. Namdeo: Formal analysis, Methodology, Resources, Conceptualization, Writing – review & editing, Supervision, Funding acquisition, UK project administration. L. Bramwell: Methodology, data Formal analysis, Writing – review & editing, Supervision. Y. Miquelajauregui: Methodology, data Formal analysis, Writing – review & editing, Supervision. C. G. Resendiz-Martinez: Data acquisition, Validation and graphics. M. Jaimes-Palomera: Data acquisition, Validation and graphics. F. Luna-Falfan: Data acquisition, Validation and graphics. A. Terrazas-Ahumada: Data acquisition, Validation and graphics. K.J. Maji: Data acquisition, Validation, writing. J. Entwistle: Writing – review & editing. J.C. Núñez Enríquez: Writing – review & editing. J.M. Mejia: Writing – review & editing. A. Portas: Writing – review & editing. L. Hayes: Writing – review & editing. R. McNally: Writing – review & editing.

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